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8	Mitigating forest biodiversity and ecosystem service losses in the era
9	of bio-based economy
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18 Abstract:

19 Forests play a crucial role in the transition towards a bioeconomy by providing biomass to 20 substitute for fossil-based materials and energy. Increasing forest harvest levels to meet the needs 21 of the bioeconomy may conflict with biodiversity protection and ecosystem services provided by 22 forests. Through an optimization framework, we examined trade-offs between increasing the 23 extraction of timber resources, and the impacts on biodiversity and non-wood ecosystem services, 24 and investigated possibilities to reconcile trade-off with changes in forest management in 17 25 landscapes in boreal forests. A diverse range of alternative forest management regimes were used. 26 The alternatives varied from set aside to continuous cover forestry and a range of management 27 options to reflect potential applications of the current management recommendations. These 28 included adjustments to the number of thinning, the timing of final felling and the method of 29 regeneration. Increasing forest harvest level to the maximum economically sustainable harvest had 30 a negative effect on the habitat suitability index, bilberry yield, deadwood diversity and carbon 31 storage. It resulted in a loss in variation among landscapes in their conservation capacity and the 32 ability to provide ecosystem services. Multi-objective optimization results showed that combining 33 different forest management regimes alleviated the negative effects of increasing harvest levels to 34 biodiversity and non-wood ecosystem services. The results indicate that careful landscape level 35 forest management planning is crucial to minimize the ecological costs of increasing harvest 36 levels.

37

38 Keywords: Bioeconomy, Trade-off analysis, ecosystem services, optimization, forest

39 management

40 Significance Statement:

41 A policy-policy conflict exists between the desire to increase the utilization of bio based 42 renewable resources and the desire to protect and conserve biodiversity. We examine and 43 evaluate the potential for these policies to be concurrently pursued. Through a case study in 44 Finland, we highlight the possibility to increase harvesting while promoting a set of biodiversity 45 and ecosystem service indicators. The impacts of increasing harvesting levels are shown on a 46 selection of both biodiversity and ecosystem service indicators. Through careful landscape level 47 forest planning, harm caused by intensifying harvests to the biodiversity and ecosystem service 48 indicators can be mitigated.

49

50 Introduction

51 In order to reduce dependence on non-renewable resources, manage natural resources sustainably, 52 mitigate and adapt to climate change, and maintain competitiveness, Europe is moving away from 53 an economy based on use of non-renewable resources and towards a bioeconomy. Forests provide 54 jobs, income and biomass for substituting fossil-based materials and energy, and compared with 55 other sources of biomass forests have the advantage of a large production potential, which does 56 not threaten food security (Ollikainen 2014; EC 2012b). Currently, the forest and wood industry 57 together with paper and pulp industry currently cover 30% annual turnover and 22% of the 58 employment in the EU bioeconomy (EC 2012b). The EU forest strategy and national bioeconomy 59 strategies and policies stress the importance of development of new wood-based materials and products (Finnish Ministry of Employment and the Economy 2014; EP 2014; Skog22 2015). In 60 61 addition, more forest biomass is needed in the energy transition to meet the renewable energy

targets (Beurskens & Hekkenberg 2011; Szabó et al. 2011; Bentsen & Felby 2012). The total energy use of biomass is expected to double from 2005 to 2020 to cover over half of the final renewable energy consumption of 10 exajoules in 2020, and over 55% of the biomass supply is predicted to come from forest (Scarlat et al. 2015). Consequently, national bioeconomy strategies relying on wood, climate and renewable energy policies together with an increasing demand for forest-based products are drivers for an increase in forest harvest levels in Europe (Mantau et al. 2010; Frank et al. 2016).

69 Intensifying biomass harvests may conflict with multiple other social economic and environmental 70 functions of forests. Forests also contribute to water quality, reduce flooding, provide recreational 71 services and non-wood products such as game, berries and mushrooms, prevent soil erosion, foster 72 biodiversity and mitigate climate change through carbon sequestration and storage (EC 2012a; 73 Nabuurs et al. 2015). Previous studies have shown trade-offs between intensifying biomass 74 harvesting and climate regulation through carbon sequestration (Schulze et al. 2012; Zanchi et al. 75 2010; Kallio et al. 2013; Triviño et al. 2015), collectable goods (Peura et al. 2016), deadwood and 76 recreational attractiveness (Verkerk et al. 2014), and maintaining high levels of biodiversity 77 (Mönkkönen et al. 2014). Therefore, bioeconomy targets aiming at intensifying biomass harvests 78 may conflict with other policy goals, such as the EU biodiversity strategy, which pursues halting 79 biodiversity loss by 2020. However, previous studies also indicate that careful forest management 80 planning may reconcile these conflicts or reduce the negative impacts (Triviño et al. 2017; Repo 81 et al. 2015), and possibly pave the way for increasing timber harvests while minimizing harm to 82 other ecosystem services.

In boreal Europe wood and forest-based products form the basis of current and future bioeconomy
(e.g. Skog22 2015; Finnish Ministry of Employment and the Economy 2014). For example, the

85 Finnish forestry, the bioeconomy currently represents 16% of the national economy and wood 86 product and pulp and paper industries cover over 40% of output and 80% of the exports of the 87 current national bioeconomy (Finnish Ministry of Employment and the Economy 2014). To boost 88 the transition towards an increased bio-based society, Finland aims to diversify wood use and to 89 increase forest harvesting to almost maximum sustainable harvest level from a timber extraction 90 perspective (Finnish Ministry of Employment and the Economy 2014; Lehtonen et al. 2016). In 91 addition to increased timber harvests, to meet the renewable energy targets agreed in the European 92 Union (EC 2009), for example Finland is aiming to triple the use of forest harvest residues, such 93 as tree tops, branches and stumps in energy production compared with the year 2009 (Ministry of 94 Employment and the Economy 2010).

95 A recent review suggests that intensive production forestry may have substantial effects on 96 numerous ecosystem services, and that these effects may be harmful or beneficial depending on 97 stakeholders (Pohjanmies et al. 2017a). Therefore, bioeconomy policy impacts on alternative 98 stakeholder groups' vary, and identifying winners and losers by evaluating the effects of 99 bioeconomy policies on alternative ecosystem functions and services will make political decision-100 making more transparent. Further, this increased intensification of forest use may promote a 101 homogenization, which may threaten biodiversity at a landscape level (Stein et al. 2014). Since 102 the Finnish forest land area covers 14% of the EU 28 countries (Peltola 2014), the effects of 103 intensifying biomass harvests on forest ecosystem services and species dependent on forests will 104 have importance on the European scale. As Sweden and Norway utilize a similar form of forest 105 management as Finland, the relevance of this study can be valid for a much greater share of 106 European forests.

107 Previous studies evaluating the transition to a forest-based bioeconomy have focused on how 108 increasing forest harvest levels impacts either the forest carbon balance, ecosystem services or 109 biodiversity The increase in timber harvests and forest harvest residue extraction rates reduce the 110 carbon stocks of biomass and soils reducing the carbon sink capacity of the forest (e.g.; Sievänen 111 et al. 2014; Frank et al. 2016). A scenario analysis in Finland to the year 2045 has shown that 112 increasing forest harvests to maximum economically sustainable harvest level reduces the forest 113 carbon sink and this sink may become an emission source if harvests are increased to the maximum 114 economically sustainable harvest level (Lehtonen et al. 2016). At a European level, a scenario 115 approach has been used to evaluate the impact on a variety of ecosystem services due to a shift in 116 policy (Verkerk et al. 2014). From a multi-objective optimization framework questions relating to 117 evaluating the sustainability of ecosystem services (ESS) and biodiversity have been addressed 118 through a direct approach (i.e. Diaz-Baltiero et al. 2016; Wam et al. 2016), or through zonation 119 techniques such as TRIAD (i.e. Montigny and MacLean 2006; Carpentier et al. 2016). Recently, 120 Heinonen et al. (2017) have conducted a scenario analysis examining the impact differing 121 harvesting intensities will have on a selection of biodiversity indicators. However, comprehensive 122 assessment of the effects of increasing forest harvest levels on different ecosystem services and 123 biodiversity are still lacking. Moreover, we do not know if and how changes in forest management 124 could minimize the possible harm resulting from increasing harvest levels to the environment.

In this study, we explore the effects of increasing forest harvest levels on biodiversity and nontimber ecosystem services. Using a comprehensive large scale dataset combined with long-term simulation of forests and multi-objective optimization tools we i) study how increasing forest harvest level affect biodiversity, non-wood products and carbon storage in boreal forests, and ii) suggest how landscape level forest planning can minimize these possible conflicts and even produce synergies. This study quantifies the effects of policies promoting increasing harvest levels on biodiversity and ecosystem services. The findings of this study can frame policy discussions on how to determine the most appropriate harvesting level and how to adapt forest management recommendations to increasing harvesting levels, taking into account a variety of environmental criteria.

135 Material and Methods:

136 To demonstrate the impact of changing the policy towards fully utilizing the maximum 137 sustainable yield (a quantity of timber products than can be harvested continuously year after 138 year), a regional level analysis is proposed. As forest industries require a stable source of raw 139 materials for production purposes, changing the quantity of timber harvested will influence the 140 ability of industry to source materials from the local region. The region under consideration was 141 comprised of 17 watersheds in central and southern Finland. The specific boundaries of the 142 watersheds were defined as third-level catchment areas, delineated by the Finnish Environment 143 Institute (SYKE 2010). The watersheds were selected to represent existing variation in overall 144 productivity (variation in soil types) and their current conservation capacity (variation in age 145 distribution). Each watershed has a differing initial state and a different productivity potential for 146 providing timber, ecosystem services (ESS) and biodiversity (BD) (for more detailed description 147 of forests in the selected watersheds, see Pohjanmies et al. 2017b). The entire region is slightly 148 over 48,770 ha and is composed of 32,276 stands (homogenous parcels of forested land). The 149 stand level data used was obtained from the local forest authority. The analysis focuses on 150 understanding how increasing the intensity of the harvests from 60% to 100% of the maximum 151 sustainable yield will impact the potential of providing other ecosystem services and maintaining 152 biodiversity. This range of harvesting intensity was selected because it encompasses the current

level (<70%) (Peltola 2014) and the targeted level according to the national policy (close to 100
%).

155 A total of five indicators were included in this analysis: timber income, habitat suitability index 156 combined for six indicator species, bilberry yield (Vaccinium myrtillus L.), carbon storage in 157 woody biomass and in soil, and deadwood diversity. Income from timber is the summation of the 158 price of the timber assortments multiplied by the quantity of the assortments. This represents the 159 monetary value of the flow of timber from the forest. Because of even-flow constraint in our 160 optimization problem (see below) discounting timber income is not needed. The price of the 161 timber is based on the assortment (i.e. saw logs or pulp wood) for each tree species, and we used 162 the average values from the recent past (Peltola 2014).

163 The ecosystem service indicators selected were the carbon storage and the bilberry yield. Carbon 164 storage was evaluated as the total carbon held within the forest. For this analysis we do not 165 consider the potential of carbon storage in the final products of the forest industry. The carbon of 166 standing timber and deadwood was evaluated as 50% of the dry biomass. Soil carbon was 167 evaluated using two models. For mineral soil the Yasso07 model were used (Liski et al. 2005, 168 Tuomi et al. 2009, 2011), and peatland soils were modeled using the carbon flux models 169 proposed by Ojanen et al. (2014). The latter provides an underestimate of the total carbon in the 170 forest, as the initial stocks of carbon in peat soils are not included but still allows evaluating the 171 changes in the soil carbon pool. The quantity of bilberries, an important non-timber product in 172 boreal forests, was calculated by the forest was predicted using the model of Miina, Hotanen and 173 Salo (2009). The bilberry models are based on empirical data, and use the site type, dominating 174 tree species, regeneration method, altitude, stand age and stand basal area as variables.

175 To evaluate the biodiversity indicators, deadwood availability and a combined habitat suitability 176 index were used. Deadwood was selected as a biodiversity indicator because in boreal 177 Fennoscandia, 20-25% of the forest-dwelling species are dependent on deadwood resource, and 178 species dependent on deadwood constitute 60% of the red-listed species (Siitonen 2001). 179 Deadwood volume is rather limited in Finnish forests, with an average of 3.8 m³/ha of deadwood 180 in Southern Finland and 8.0 m³/ha of deadwood in Northern Finland (Peltola 2014), which is 181 considerably less than in natural forests where the reported average volumes range from 20 182 m3/ha on infertile forest types to 120 m3/ha on more productive sites (Siitonen 2001). Since the 183 deadwood dependent species have specific requirements for deadwood quality (e.g. Tikkanen et 184 al. 2007), in this study deadwood availability was a function of total deadwood volume 185 multiplied by the diversity of deadwood. Diversity, scaling between 0 and 1, was calculated as 186 the volume of deadwood in different tree species, decay stage and diameter classes by the inverse 187 of Simpson's diversity index (Triviño et al. 2017). Thus, a stand will have high deadwood 188 availability if it contained large total volume divided evenly across different deadwood classes. 189 The combined habitat suitability index was evaluated as the combination of six habitat suitability 190 indices. The habitat suitability of Capercaillie, hazel grouse, three-toed woodpecker, lesser-191 spotted woodpecker, long-tailed tit and Siberian flying squirrel (Mönkkönen et al. 2014) were 192 integrated through a multiplicative approach (Triviño et al. 2017). These species were selected 193 to represent a wide range of habitat types as well as social and economic values including game 194 birds, umbrella and threatened species. Species-specific habitat suitability index (HSI) varies 195 between 0 (unsuitable habitat) and 1 (most suitable habitat) and is related to the probability of the 196 presence of the species in the stand. We thus calculated a combined HSI for the six species as the 197 combined probability of independent events:

198 $HSI_c = 1 - \prod_{i=1}^{6} (1 - HSI_i)$

The combined HSI is related to the probability that at least one of the species is present, and returns a high value for a stand if at least one of the species has high HSI, and a value close to zero if a stand provides low suitability for all the species.

202 The initial forest data was provided by the Finnish Forest Center. The data is comprised of stand level forest information, with a description of the stand level characteristics and information on 203 204 the strata which compose the forest stand. The stands have a median area of 0.98 ha, with a 205 minimum area of 0.01 ha and a maximum area of 61.79 ha. The forest is inventoried through 206 remote sensing technology (Airborne Lidar Scanning; Næsett 2007), and is updated in a 10 year 207 cycle. Predictions of the future forest states were made through the use of a forest simulator 208 (SIMO; Rasinmäki et al. 2009). SIMO is an adaptive simulation open-source framework 209 designed specifically for forest management planning. The modelling framework consists of over 210 400 equations to predict, among other things, the growth of the diameter and height of each tree 211 and the probability of a tree death. For the majority of the management regimes, the prediction of 212 the development of the forest stand was conducted using the forest models of Hynynen et al. 213 (2002). One management regime (continuous cover forestry, CCF) used the Hynynen et al. 214 (2002) models to predict the forest stand development until the point in time where harvesting 215 actions occurred, and converted the stand to a CCF stand. Following conversion to a CCF stand, 216 the continued development of the forest stand was predicted using the models by Pukkala et al. 217 (2013). This was done as the models of Hynynen et al. (2002) are specific to even-aged forests, 218 and the models of Pukkala et al. (2013) are developed for uneven-aged forests. A time horizon of 219 100 years was selected, divided into 20 periods each 5 years long. The length of the time horizon 220 was selected to examine what may happen over an entire rotation period (the length of time

required for a seedling to grow into a harvestable tree). This choice was made to ensure that theharvest level could be kept constant for the continued sequence of rotation periods.

223 Management regimes were created to reflect potential decisions that forest owners may make 224 over the time horizon. A total of 19 management regimes were used to represent how the forest 225 may be managed. One management regime for all stands was to set aside (SA), and simply allow 226 the stand to grow. A second alternative was to conduct continuous cover forestry (CCF), where 227 periodically large trees are removed, and growth and regeneration is left to nature. The 228 remaining alternatives were modifications of conducting business as usual (BAU). Starting from 229 bare ground, the management regime starts with a selection of pre-commercial actions was taken 230 to promote forest growth, followed by possible commercial thinnings and final felling to extract 231 timber. Modifications were created by restricting the number of thinnings, by adjusting the 232 timing of final felling, and by switching from artificial regeneration to natural regeneration. A 233 more detailed description of the management regimes can be found in the supplementary 234 material (Appendix S1).

To examine a variety of potential scenarios, we utilize a theoretical landscape level planning approach, where all decisions are taken at an individual stand level. From a conservation perspective, species persistence primarily depends upon habitat availability at the landscape/regional scale (Fahrig 2017). Thus, we focus on examining the trade-offs between harvesting actions and habitat availability of forest indicator species, i.e. areas of less intensively managed forests at a landscape scale.

Once the stands have been predicted for the feasible management regimes, optimization methods
were used to evaluate the maximum possible periodic harvest. This is an even-flow problem,

243 where each period has a similar quantity of timber flowing from the forest to the consumers. This

is a common problem in forestry, as pulp and timber mills require a relatively constant flow of

245 inputs to enable continual production. The optimization model can be framed as a linear

246 programming problem (Johnson & Scheurmann 1997):

247 Model 1:

248 **[1] max**
$$z = \sum_{k=1}^{K} \sum_{j=1}^{J_k} c_{kj1} x_{kj}$$

249 Subject to:

250 **[2]**
$$\sum_{k=1}^{K} \sum_{j=1}^{J_k} c_{kj1} x_{kj} \le \sum_{k=1}^{K} \sum_{j=1}^{J_k} c_{kjt} x_{kj}, t = 2, ..., T$$

251 **[3]**
$$\sum_{j=1}^{J_k} x_{kj} = \mathbf{1}, k = \mathbf{1}, \dots, J$$

252 [4]
$$x_{kj} \ge 0 \forall k = 1, ..., K, j = 1, ..., J_k$$

where z is the objective function value, c_{kjt} is the value of the timber available from stand k according to management regime j at the t^{th} period, x_{kj} is the decision for stand k to conduct management regime j, K is the total number of stands under consideration, J_k is the total number of management regimes for stand k, and T is the total number of periods under consideration. In this linear programming model, the objective function is to maximize the first period timber flows, while the constraint detailed in [2] ensures that all future periods can provide at least as much timber flow as what was obtained in the first period. Constraint [3] ensures that each stand is assigned some management regime and [4] is a non-negativity constraint, ensuring that thedecisions for assigning management regimes are always positive (or zero).

262 The objective value of the previous model highlighted the maximum even-flow of the value of 263 timber and does not actively consider the optimization of any other indicators. A second model 264 was developed to analyze the trade-off between the even-flow requirement and a selection of 265 four provisioning and conservation indicators. To accomplish this, a compromise programming 266 formulation was used (Yu 1973). Compromise programming allows for selecting the most 267 appropriate distance metric from L^p space, and relates to other multi-objective programming 268 methods (Tamiz et al. 1998, Romero et al. 1998, Cisneros et al. 2011). When the distance metric 269 $L^{p} = 1$, the focus is on minimizing the aggregated sum of the deviations, while the distance 270 metric $L^{p} = \infty$ focuses on minimizing the maximum sum of the deviations. For this study, we use 271 the distance metric $L^{p} = 1$, assuming equal weights for all objectives However, another metric 272 might be equally valid depending on the preferences of the decision maker. The objective 273 function was to minimize the weighted normalized difference from the ideal and nadir values, 274 while ensuring that the timber provided by the plan meets a specific percentage of the theoretical 275 maximum even-flow found in the previous model. This provided a method of evaluating the 276 trade-offs between increasing the amount of timber harvested and the impacts on the ecosystem 277 services.

278 Model 2:

279 **[5]** min
$$I = \left(\sum_{e=1}^{E} w_e^p \left| \frac{d_e^* - y_e}{d_e^* - d_{e^*}} \right|^p \right)^{1/p}$$

280

281 Subject to

282 **[6]**
$$\sum_{t=1}^{T} \sum_{k=1}^{K} \sum_{j=1}^{J_k} d_{ekjt} x_{kj} = y_{ej} \ e \in E$$

283 **[7]**
$$\sum_{k=1}^{K} \sum_{j=1}^{J_k} c_{kj1} x_{kj} \ge z * f$$

and [2], [3] and [4].

285 where d_{ekjt} is the value of the ecosystem service or biodiversity indicator value e available from stand k according to management regime j at the t^{th} period, w_e is the preferential weight assigned 286 287 to criterion e, while d_e^* and d_{e*} are the ideal and anti-ideal values for criterion e.[8] Parameter f is 288 set to determine the percentage of maximum periodic timber harvest. The trade-off between the 289 set of ecosystem services and biodiversity indicator values in the objective function and the 290 timber required can be evaluated by modifying this parameter. The objective function [5] 291 minimizes the weighted normalized distance for all criteria under consideration. As presented, 292 this is a non-linear model, so prior to solving, a conversion to a linear format eases the 293 computational difficulties, for specific techniques to accomplish this readers are referred to 294 Tamiz et al. 1998. Constraint [6] calculates the ecosystem services and biodiversity values for a 295 specific decision, and constraint [7] requires that a specific flow of timber is met for each time 296 period. To summarize, in this model, the objective function was to maximize a set of ecosystem 297 services and biodiversity indicators while the constraints ensure a steady flow of timber for all 298 periods under consideration.

To find the ideal and anti-ideal values (d_e^* and d_{e*}), the following simple linear programming model was used:

301 **[8]** max
$$d_e^*$$
 or min $d_{e*} = \sum_{t=1}^T \sum_{k=1}^K \sum_{j=1}^{J_k} d_{ekjt} x_{kj}$

302 subject to [3] and [4].

303 To highlight the importance of planning for all indicators of interest, we examined the range of 304 solutions possible if the focus was only on the requirement of sustaining an even-flow of timber 305 resources. If the only indicator of interest is the even-flow of timber, as the requirement for 306 maximum even-flow is decreased, additional options of achieving the specific levels of timber 307 were possible. To evaluate the expected result of the ecosystem services and biodiversity 308 indicators, we enumerated a large sample of possible solutions. The solutions were created with 309 an aim to be evenly distributed amongst the possible outcomes. A detailed description of how these solutions were created can be found in the supplementary material (Appendix S2). 310

311 Results

312 Increasing forest harvest level to the maximum economically sustainable harvest will have a 313 negative effect on biodiversity and non-timber ecosystem services even when management was 314 optimized to meet alternative objectives (Figure 1). Maximizing harvest level is particularly 315 detrimental to biodiversity indicators. If 100% of the maximum sustainable yield was harvested 316 the deadwood availability decreased 70% and combined habitat availability by 26% compared to 317 when focusing the sustainable yield to 60% of the maximum. Losses for ecosystem service 318 indicators were more moderate: 30% decline in bilberry yield, and 12% in carbon storage (Figure 319 1). The losses in carbon storage showed a rather linear decline with increasing harvest level. For

bilberry yield, the timber harvest level can increase up to some 85-90% of the maximum
sustainable harvest level without substantial negative impacts. As the harvest levels increased the
habitat availability and deadwood diversity indicator values of the studied 17 watersheds
converge (Figure 1, grey and red lines). This suggests a loss of landscape specific biodiversity
characteristics.

325 The results above were based on multi-objective optimization, i.e. are the highest achievable 326 levels of biodiversity and ecosystem service indicators at different levels of timber harvesting, 327 and achieving them requires careful planning. For this analysis, we assumed equal importance 328 between objectives. However, this assumption may be relaxed through integration of stakeholder 329 preferences. This can be done e.g. through interactive multiobjective optimization, where 330 stakeholders are allowed to gain an understanding of the decision problem and provide 331 preferences throughout the process. (Miettinen, 1999; Miettinen and Ruiz 2016). To examine the 332 importance of setting appropriate weights, a payoff table highlighting the best and worst cases 333 for each indicator at each harvesting level is provided in Appendix S3. As the harvesting 334 requirement is reduced, the range of optimal solutions increases, highlighting how at the 335 landscape level preferences (i.e. regional planners) can influence the optimal solution. If careful 336 planning is not done considerable losses in non-timber benefits accrue in almost all cases (Figure 337 1, dashed blue line). Only at the maximum level of timber harvesting level, all of the solutions 338 are rather similar, and consequently, there is very little flexibility for planning (Figure 1 & 2). 339 Planning benefits are particularly large for deadwood availability, as there is a loss of nearly half 340 of deadwood diversity due to timber harvesting incurred without planning at 60% level of 341 timber-flow (Figure 2).

342 The distributions of the optimal stand specific management regimes are different for different 343 harvest levels (Figure 3). At the lowest harvest levels (60% of the maximum), the management is 344 dominated by three regimes: SA (38%), CCF (42%) and a version of BAU with green tree retention (13%). Together these three regimes account for 93% of the total area. The remaining 345 346 management regimes were applied to the remaining area, however none were applied to more 347 than 2% of the management of the entire region. The stands assigned to the SA regime consisted 348 of a range of initial conditions. For the 60% harvest level, the SA regimes had an initial average 349 of 188 m3/ha of timber and an average age of 59 years, compared to 149 m3/ha and 47 years for 350 the general initial conditions. Alternatively, when the requirement for timber flow is the 351 maximum sustainable harvest level, seven management regimes account for 91% of the total area 352 with the continuous cover forestry (41%) being the most prominent regime. At this harvest level, 353 the possibility to set aside the forest is limited, and the harm is minimized by a diverse set of 354 clear-cut based management regimes with varying rotation lengths and thinning levels, as well as 355 with a frequent use of continuous cover forestry.

356 Discussion

357 Our results show that focusing a strategy of increased timber flow will likely result in 358 considerable losses in biodiversity and ecosystem services, and consequently produce ecological 359 and social costs. Ecological costs are particularly pronounced as the indicators are shown to 360 decrease >30% compared to what is achievable at the current timber harvest levels. At current 361 harvest levels biodiversity is already threatened due to intensive forestry reducing characteristics, 362 resources and variation that are important for forest species (Hanski 2000). Deadwood stocks in production forests of southern Finland are ~3-4 m³/ha; for more demanding deadwood associated 363 364 species to occur a level of 20 m³/ha is required (e.g. Junninen & Komonen 2011). At current

harvesting levels, the expected deadwood availability values for our study region correspond
rather well to measured values. Thus, the projected 70% decrease in deadwood availability is
realistic and would further shift the quality of forests away from the ecological sustainable level
of deadwood resources. Therefore, pursuing the bioeconomy policy will further increase species
endangerment, for forest-associated species in general and deadwood dependent species in
particular.

371 Our results also indicate that by increasing the level of harvesting there will be a loss of variation 372 between landscapes, which initially differed in their ability to provide non-timber ecosystem 373 services and biodiversity. In other words, landscapes with a poor biodiversity values at current 374 harvest levels (<70%) remain poor, while highly biodiverse landscapes also become poor. This 375 convergence among landscapes occurs because with increasing harvest level, harvesting actions 376 are conducted in stands with progressively higher biodiversity values. The convergence reduces 377 environmental heterogeneity at a regional scale, which is a further threat to biodiversity. There is 378 strong evidence that environmental heterogeneity is an important universal driver of biodiversity 379 at landscape to global extents (Stein et al. 2014).

380 For this study, we did not include potential climate change impacts into the growth models, so 381 the results may be an under/over estimation of the different ecosystem services. For instance, in 382 Finland, increased temperatures could positively impact forest growth and tree mortality. This 383 would simultaneously increase deadwood decomposition resulting in a faster turnover rate of 384 deadwood resources (Mazziotta et al. 2014) and a larger proportion of deadwood associated 385 species losing habitats than gaining more habitat (Mazziotta et al. 2016). The result would be 386 positive from a timber extraction point of view, but negative from a biodiversity perspective 387 supporting the findings of earlier studies (Schulze et al. 2012; Sievänen et al. 2014). Forest

388 management changes, which increase forest carbon stocks, such as fertilization, could possibly 389 partly compensate for the forest carbon loss. However, fertilization raises other environmental 390 concerns. Thus, increased harvest level will have a direct negative effect but likely also an 391 indirect negative effect, via climate change, on biodiversity.

392 Additionally, we did not study the impacts of other possible sources of uncertainty. The 393 development of forest resources was predicted through the use of growth models. These models 394 are based on sets of assumptions and as with all forecasts the future cannot be predicted without 395 error (Diebold 2001). The possibility exists to include these sources of uncertainty in the 396 optimization framework through stochastic programming (Birge & Louveux 2011). Through a 397 stochastic framework, questions related to the distribution of the indicators can be examined. 398 However, currently the computational cost to execute such a framework on this problem is 399 exceptionally high. For the question related to the policy of implementing higher sustainable 400 yields, uncertainties need not be explicitly included in the framework, rather the possible impacts 401 should be discussed.

402 In this study, the estimate of carbon stored in the forest is an underestimate, as the initial state of 403 carbon stored in the peat is not included in the analysis. This was due to a lack of precise data 404 regarding the quantity of peat for the large area under consideration. In this study, a total of 15% 405 of the area was forested peatlands, which could reflect a store of carbon of 3,600 kt C (using 406 estimates of 500 t C/ha) (Minkkinen & Laine 1998; Turunen 2008). As this study is essentially 407 interested in the amount of carbon sequestered (where this change can be seen through the 408 fluxes), incorporating the initial state of stored carbon from the peat lands will not impact the 409 results of this study.

410 This study focused on the use of providing a steady amount of timber resources from the forest over a long period of time. This concept has been a feature of sustainable forestry since the early 411 412 18th century (von Carlowitz 1713). This requirement to provide a continuous timber supply is an 413 economic sustainability requirement, which prevents excessive destruction to the forests. 414 However, while the forests may provide a constant flow of timber, various other issues of 415 sustainability, such as sustained provision of collectable forest products, or maintenance of 416 biodiversity, are ignored with this approach. For biodiversity, persistence in time of species is critical because global extinctions are irreversible and regional extinctions maybe time-417 418 consuming to remedy given the sparsity of source populations in production forest landscapes 419 (Hanski 2000). Thus, sustained availability of habitats and even flow of resources for species are 420 critical. From the bioeconomy perspective, the supply of each specific biomass type may require 421 a sustainable flow (Ollikainen 2014), so the realm of sustainability should be opened up and 422 include various economic and ecological aspects of sustainability.

423 The potential exists to increase the timber harvest level while limiting the negative impacts on 424 ecosystem services and biodiversity indicators. In this study, this potential was evaluated through 425 optimization, and implementation would require careful planning. Relative benefits from 426 planning are generally high but varied among the indicators (Fig 2). Careful landscape level 427 planning can offer a means to reduce the negative effects of increasing forest harvest levels on 428 biodiversity and ecosystem services. In this study, a failure in implementation of optimal 429 landscape level plans resulted in a loss of 30-40% in ecosystem service and biodiversity 430 indicators at most timber harvesting levels (Figure 1). Thus, to limit the losses of the potential of 431 landscapes to maintain biodiversity and ecosystem services careful planning will become 432 increasingly important in the era of bioeconomy. How to successfully conduct this planning

433 should be aided through an exploration of historical development of forest resources. For 434 instance, Angelstam et al. (2018) and Naumov et al. (2018) have explored the competition of 435 biodiversity and timber production through a spatial comparison of countries with different 436 historical development of forest use. Ideally, resource harvesting should be targeted to sites with 437 the highest timber production potential and cause the smallest losses to biodiversity and 438 ecosystem services. Correspondingly, resources for nature conservation should be invested to 439 maintaining non-timber ecosystem service provisioning in areas with high ecological and social 440 values but low timber production potential. Kareksela et al. (2013) coined this as negative impact 441 avoidance approach and successfully applied this to land use planning for peat mining. 442 But mere planning is not enough; plans need to be implemented. In a forestry context, this will 443 require involvement of and acceptance by forest owners. If only a proportion of stakeholders 444 ignore the suggested management plan, inefficiencies will be introduced. In practice, conducting 445 careful landscape level planning is difficult to accomplish, as the forest properties are controlled 446 by a large variety of stakeholders with differing intentions and objectives (Eriksson and Hammer 447 2006; Angelstam et al. 2011). Some commodities such as timber are considered private property, 448 benefiting primarily the landowner while others are considered public goods. For example, 449 climate change mitigation provides a global benefit by reducing atmospheric CO2 levels, while 450 water quality regulation, and recreational use, natural collectable products (e.g., berries and 451 mushrooms) profit mostly the local community. Private landowners typically lack the incentive 452 to manage land to provide ecosystem services and biodiversity conservation benefits in cases 453 where the benefits produced on their land accrue to others.

However, aggregating forest planning for even a small set of forest holdings can mitigate the
trade-off between increasing forest harvest levels. For example, e.g. Pohjanmies et al. (2017b)

456 observed that approximately 100 stands or 200 ha, i.e. less than ten owners, is large enough to 457 effectively mitigate the conflict between timber production and carbon storage. Thus, 458 incentivising forest owner's collaboration to landscape level planning may not be an impossible 459 mission. Because priorities between forest owner level planning and landscape and regional level 460 forest planning are often mismatched, the implementation of the landscape level plan incurs costs 461 and benefits unevenly among forest owners. To align the priorities, policy tools, such as 462 monetary compensation for voluntary conservation (e.g. METSO 2008), could compensate for 463 losses to those forest owners who face large private costs for providing common goods in terms 464 of biodiversity and non-timber ecosystem services. One way to differentiate landscapes where 465 environmental and social objectives have priority from timber production landscapes in regional 466 forest resource management planning are systematic zoning tools, such as the inverse spatial 467 prioritization (Kareksela et al. 2013). Zoning, together with incentives and monetary 468 compensations to forest owners for extra planning work, and economic losses could improve the 469 protection of public interests in boreal production forests in the era of bioeconomy. 470 In the era of bio-economy ensuring ecological social and economic sustainably of boreal forest 471 management requires, in addition to careful planning, diversification of management regimes. 472 We found that at most levels of timber harvesting, optimal management is dominated by set-473 asides and continuous cover forestry, and clear-cut based forestry becomes the prevailing - but 474 not exclusive – management regime only at very high levels of timber harvesting (>95%). Thus, 475 by relying on the application of the standard practice of final felling by clear-cuts results in costs 476 for economic, ecological and social aspects. These results are similar to earlier literature, where

477 continuous cover forestry is shown to often be better in providing timber and non-timber

478 ecosystem services than clear-cut forestry (Pukkala et al.2011; Pukkala 2016; Tahvonen 2016;
479 Tahvonen & Rämö 2016; Peura et al. 2018).

480 Some forest certification programs (e.g, FSC) require setting aside a minimum of 5% of forest 481 area. Our results suggest that optimal set-aside level is much higher at most levels of timber 482 harvesting, e.g. more than 25% currently (at <70% harvest level), and 8% at 90% timber harvest 483 level. Thus, it is optimal to concentrate forest harvesting to sites where yields are highest and 484 losses to biodiversity and non-timber ecosystem services lowest, allowing for large areas of 485 forests to be set aside. Currently, around 2% of forest area is formally protected in south boreal 486 Fennoscandia, with an estimated 6-7% of the forested area protected both formally and 487 voluntarily (Peltola 2014; Angelstam et al. 2011) and therefore, more investments in forest 488 protection are optimal and possible even with increasing timber requirements.

489 Economic growth and the shift from non-renewable resources is a very understandable 490 justification for EU and national level strategies to promote increased extraction of timber 491 resources. However, this focus should link to other international, EU level and national level 492 policy agreements that aim at halting biodiversity loss and maintain ecosystem services. A recent 493 EU Parliament resolution (EU Parliament 2016) urges for considerable additional efforts for 494 biodiversity protection in European forests. Likewise, the international Strategic Plan for 495 Biodiversity 2011-2020 (Aichi Biodiversity Targets) requires that by 2020 all areas under 496 forestry are managed sustainably ensuring the conservation of biodiversity, 17 per cent of 497 terrestrial area is conserved through effectively and equitably managed, well-connected protected 498 areas and other effective area-based conservation measures, and ecosystems that provide 499 essential services are restored and safeguarded. Our results indicate that the Finnish forest 500 strategy (i.e. achieving maximal sustainable timber harvest level) as well as EU level and

501 national bioeconomy policies (targeting considerable increases in forest harvesting) are in 502 conflict with the biodiversity and ecosystem services policies, i.e. there is a policy-policy gap. 503 Policy analysis identifies this ignorance of goal conflicts in Finnish forest policies (Makkonen et 504 al. 2015; Kröger & Raitio 2016). Disintegrated, sectoral policies are ineffective and 505 unsustainable (Winkel & Sotirov 2016), and better policy coherence is therefore desirable. Our 506 results show that to bridge the policy-policy gap in forest use in practice, a multi-objective 507 planning approach is needed where economic objectives are neatly balanced with environmental 508 and social values.

509 Conclusions

510 Increasing the requirement for resource extraction from natural resources will require an 511 appropriate balance between economic, ecological and social objectives, possible with careful 512 multi-objective planning. In boreal forests, the diversification of management regimes will be 513 needed for overall sustainability, and a shift from clear-cut forestry would provide considerable 514 benefits for forest owners and the society. Our results indicate that careful forest planning can 515 reduce the negative effects of increasing forest harvest levels on biodiversity and ecosystem 516 services.

From practical perspective, a viable solution would be landscape sparing, i.e. spatially segregating landscape where timber production is the main objective from landscape with a better balance between objectives. Even though in general the effects of fragmentation are much weaker than the effects of habitat loss on a wide range of ecological responses (Fahrig 2017) ecological research has concluded that if a limited area of species habitats can be protected they should be protected in spatially aggregated clusters rather than as randomly scattered fragments.

523	This will generally reduce species extinction risk and increase the conservation benefits for a
524	given total area protected (Hanski 2011). Also from the mere human perspective it may well be
525	reasonable to aggregate efforts because, for example, larger tracks of mature forests can be found
526	more appealing for recreation than an equal area in small fragments.
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Figure 1. The combined results of all harvest levels considered. All indicators are evaluated as a per hectare. With a steady increase requirement for timber flow there is a rapid decline in the average levels of other indicators (black line). Additionally, as timber flow is increased there is convergence in values of habitat suitability and deadwood diversity between watersheds (light grey lines). This can be seen as a narrowing in the standard deviation (thick red lines). The expected solution when ecosystem services and biodiversity are not included in the optimization is shown with the dashed blue line.



Figure 2. The benefits of ensuring proper planning at different levels of even timber-flow in

terms of the ratio of the values for the optimized solution and the expected result where

762 biodiversity indicators and ecosystem services were not included in the objective function.





Figure 3. Change in the area managed according to the different regimes when there is an increasing requirement for even-flow of timber in the optimized solutions. BAU refers to alternative clear-cut based management regimes with variable thinning intensities and rotation lengths (GTR = green tree retention, w thin = with thinnings before clear felling, wo thin = with no thinnings). CCF refers to continuous cover forestry with not final felling by clear-cut, and set aside denotes permanent protection (no management). For description of management regimes see Appendix S1.